Global Ecology and Conservation 6 (2016) 299–307

Contents lists available at ScienceDirect

Global Ecology and Conservation

journal homepage: www.elsevier.com/locate/gecco

Short communication

Lack of evidence for short-term structural changes in bird assemblages breeding in Mediterranean mosaics moderately perforated by a wind farm

C. Battisti [a,*], L. Fortunati [a], V. Ferri [a], D. Dallari [b], G. Lucatello [b]

[a] Studio Natura Arcadia, via Valverde, 4, 01016 Tarquinia, Viterbo, Italy

[b] Agrifolia, Environment, Agriculture and Landscape, via Val Trompia, 108, 00141 Rome, Italy

HIGHLIGHTS

- Wind farm turbines induce a habitat perforation in oak mosaics.
- About 10 % in habitat perforation is not enough to change structure in bird assemblages.
- A lack of changes is also evident at guild (edge vs. forest species) and species level.
- Further research is necessary to search a higher threshold in habitat perforation inducing a change in assemblage structure.

ARTICLE INFO

Article history:
Received 21 February 2016
Received in revised form 26 March 2016
Accepted 27 March 2016
Available online 13 April 2016

Keywords:
Heterogeneity
Habitat perforation
Wind farms
Indirect impact
Diversity/dominance diagrams
Common species

ABSTRACT

We studied a set of common breeding birds living in a heterogeneous oak wood mosaic of Apennines (central Italy) where a wind farm occurred. Aim to assess differences in composition and structure between a treatment area (with wind farm turbines) and a control area (without wind farm turbines). We did not observe differences at assemblage (uni-and bi-variate metrics of diversity: mean species richness, Shannon–Wiener diversity index, evenness, Whittaker β_w index and diversity/dominance diagrams), guild and species level (relative frequencies). The limited habitat perforation and dissection induced by wind farm turbines and service roads (10% in area) and the consequent changes in spatial heterogeneity and level of anthropogenic disturbance (induced by a higher motor-car and people frequentation) did not seem to affect the breeding bird communities in oak mosaics, as supported also by the diversity/dominance analysis. However, our preliminary conclusions are limited only to the indirect impact on common breeding bird species and are not related on to possible direct impacts deriving from wind farm facilities and related infrastructures (e.g., direct impact for collision). Moreover, further research is necessary to detect possible higher thresholds in habitat perforation that may induce changes in breeding bird assemblages.

1. Introduction

Anthropogenic and natural spatial heterogeneity constitutes a driving force that explains patterns of species richness and abundance in animal communities occurring in patchy landscapes (Fahrig et al., 2011). Among birds, the role of fine-grained

* Correspondence to: "Torre Flavia" LTER (Long Term Ecological Research) Station, Città Metropolitana di Roma Capitale and Department of Science, University of Rome III, viale Marconi, 446, 00146 Rome, Italy.
E-mail address: c.battisti@cittametropolitanaroma.gov.it (C. Battisti).

http://dx.doi.org/10.1016/j.gecco.2016.03.012

small-scale (i.e. patch-scale) habitat disturbances on avian assemblages have been widely investigated, mainly in temperate boreal forests (e.g. Derleth et al., 1989; Forsman et al., 2010).

In mountainous and hilly areas of Mediterranean basin, historical and recent anthropogenic disturbances contributed to structuralize complex landscape patterns (Forman, 1995; Blondel and Aronson, 1999; Zamora et al., 2007 and Battisti and Fanelli, 2011). These landscapes are frequently characterized by low level of anthropization and shows a natural and human-induced patchiness (e.g. Farina, 1997 and Brotons et al., 2005). Moreover, these hilly mosaics are often located in favourable conditions concerning dominant regional and local winds, therefore many ridges and exposed sides were selected as sites to locate wind farm facilities (Williams et al., 2001; Noguera et al., 2010).

Wind farm facilities are infrastructures that may structurally and functionally affect a large set of local abiotic and biotic components at multiple spatial and temporal scales (e.g.: Winkelman, 1992, 1995; Martì and Barrios, 1995; Richardson, 2000; Langston and Pullan, 2003; Zieliński et al., 2009 and Gove et al., 2013). The impact (direct, indirect or potential) on the biotic components has been analysed for birds and bats in particular, showing a strong context- and species-specific response (Jain et al., 2010; Huso, 2011; Johnson and Erickson, 2010; Ferrer et al., 2011 and Langston and Pullan, 2003). Particularly, wind farm-related infrastructures (windmills, roads, electric lines, etc.) and related disturbances (people frequentation, motor-car transit, etc.) may also indirectly impact the population density, as they change the level both of landscape heterogeneity, so consequently disrupting the structure at assemblage and species level (Langston and Pullan, 2003).

At least for Mediterranean ecosystems, research focusing on the indirect impact at bird assemblage level are still rare (e.g. Battisti et al., 2014a, b).

Here, we report a study carried out on common breeding bird assemblages living in a poorly studied heterogeneous landscape mosaic of Apennines (Abruzzo, central Italy) where a cluster of windmills is located. Particularly, the aim of this study is to quantify the differences at bird assemblage and species level between two oak mosaics with different wind farm-induced heterogeneity, assessing the differences between a treatment area (with a wind farm) and a control area (a comparable mosaic where wind farm infrastructures are absent). Our prediction is that as wind farm infrastructures created small but well distributed open areas inside the oak mosaics, thus increasing the internal patchy heterogeneity and small-scale disturbances, we should have sampled a change in some uni- and bi-variate metrics at level of structural assemblage, guilds and species of common breeding birds.

2. Materials and methods

2.1. Study area

The study has been carried out in a natural and human-induced heterogeneous landscape near San Giovanni Lipioni (Chieti) in the Southern Abruzzo (monti Frentani, central Apennines, Italy) on a surface of about 600 ha near the river Trigno at an altitude ranging from 400 to 740 m a.s.l. (geographic coordinates: 41° 84′73″ N, 14° 56′27″ E). In this area oak patchy mosaics (MOS) are present, characterized by termophilous woods of deciduous oaks (*Quercion pubescentis-petraea*, with *Quercus pubescens* dominant and secondarily *Quercus cerris, Acer* sp., *Carpinus* sp., *Castanea sativa*) on arid calcareous and muddy soils (EU habitat type 91H0: "Pannonian woods with *Quercus pubescens*", largely diffused in central–Southern Apennines: Petrella et al., 2005). Two plant associations occur in this context: *Rosa sempervirens-Quercetum pubescentis* and *Cytiso sessilifolii-Quercetum pubescentis* (Pirone, 1995). In the surrounding, open habitats characterized by large extension of crop and uncultivated lands (wheat and secondarily vegetable crops, vineyards and olive groves) with rare shrubs, hedge and trees (mainly *Prunus* sp., *Rosa* sp., *Crataeugus* sp.) are also present. Locally a patchy mesophilous and hygrophilous vegetation also occur with *Populus* sp., *Salix* sp., *Ulmus* sp., *Acer campestris*, *Fraxinus ornus* (Pirone, 1995).

Inside the MOS habitat type, five wind turbines (coded from WTG 1 to WTG 5; Eolica Lucana—Iberdrola) have been located (Caccavone ridge, Vernone hill) between 636 and 703 m a.s.l. so perforating an 300 hectares-wide area. At landscape level, the building of windmills and their related infrastructures regarded an area of about 30 ha (6 ha of transformed area for each wind turbine), corresponding to about 10% of the MOSs where the wind farm facilities have been located. Wind turbines were placed in 2012 and started their full-time activity in winter 2013.

2.2. Protocol

To study the structure of breeding bird assemblages, in MOS we carried out an intensive surveys sessions in 2013 from 4 May to 15 June carrying out a standardized quantitative point count method (hereafter, PCM; Koskimies and Väisänen, 1991; Bibby et al., 2000 and Sutherland, 2006). The point count method has been widely used to characterize bird assemblages in terms of species richness, diversity and composition at landscape level (Sutherland, 2006). Moreover, this method provided the analysis of fine-grained bird–habitat relationships (Morrison, 2002).

In each randomly selected point count ($n = 54$), we sampled any territorial sighting or singing individual of any bird species inside a standardized distance from the observer (100 m as fixed radium of detection; see Bibby et al., 2000). Time of sampling in each point count was 5 min. When highly replicated at landscape level, this timing can provide a good description of the breeding bird assemblages (Sutherland, 2006). However, some rarer species of the breeding assemblage under study, including some diurnal raptors, woodpeckers, and rare and less detectable passerines, could be underestimated

with this duration (Sorace et al., 2000). Therefore, we consider our data only related to "common species", i.e. species locally abundant and easily detectable with the PCM method (Bibby et al., 2000; Sutherland, 2006).

To detect breeding birds, we carried out the sampling in spring season, mainly in the morning (from 6.00 to 12.00 a.m.) and always in sunny (or moderately cloudy) conditions. We maintained a minimum distance between points of 300 m to reduce biases due to double-counting of the same individuals (bias for pseudo-replication and lack of independence; Battisti et al., 2014a, b). To determine the location and distance of each point we used a GPS E-trex.

To assess differences in structure between assemblages in MOS with wind turbines and in MOS without ones, we stratified the sampling design in two sub-habitat types:

- a treatment area (MOSt), i.e. oak mosaics with windmills and relative infrastructures (5 turbines, related roads, etc.; 21 point counts). A moderately disturbance (by noise) occurred in MOSt area: a road (recently built: 2011–2012), linking the wind turbines, was periodically used from Iberdrola's technicians and practitioners (about 1–5 motor-vehicles/day);
- a control area (MOSc), i.e. a comparable oak mosaic area without wind farm and related infrastructures distant from 1.5 to 3 km from the treatment area. In the MOSc motor-vehicles never occurred due to the only presence of wood trails rarely frequented by hunters and other local people (33 point counts).

2.3. Data analyses

First, we obtained a check-list of species each one characterized by the level of conservation concern (inclusion in red list: Rondinini et al., 2013) and Bird Directive 409/79/CEE (BirdLife International, 2004).

At level of single species, we obtained their normalized relative frequency fr_i (i.e. the ratio between the number of individuals detected and the total number of individuals; ranging from 0 to 1) in total and both the two sub-habitats.

To assess an effect of wind farm perforation at guild level, we subdivided the species in two coarse-grained habitat-related guilds (Table 1): (i) open/edge species, mainly linked to open habitats and edge conditions and, (ii) mosaic/forest species, mainly linked to forest or woody mosaic conditions (data on the ecological preference of the various species were gathered from the peer-reviewed literature: Moore and Hooper, 1975; Cieslak, 1985; Opdam et al., 1985; Møller, 1987; Hinsley et al., 1995; Bellamy et al., 1996 and Ukmar et al., 2007). Then, we obtained the total values of total fr_i for each guild.

At assemblage level, for each sampling point we obtained the number of species (Sp); this parameter corresponded, at spatial level, to an α-diversity measure (i.e., the number of species referred to a single sampling point, Whittaker, 1972 and Magurran, 2004). For each sub-habitat type, at level of the whole set of point counts, we obtained the following community parameters: (i) total number of species (S), a measure of γ-diversity for the relative habitat type or sub-type (Whittaker, 1972; see also Magurran, 2004); (ii) mean number of species (a normalized metric of richness; S_{mean}), as the ratio between the number of species identified in each point count and the number of point counts; (iii) mean abundance index (a normalized metric of abundance; ABB_{mean}), as the ratio between the individuals sampled in each point count and the number of point counts. We did not account for the different detectability of the species: therefore, we obtained only a simple index of abundance.

To assess the structural complexity of each assemblage we calculated the Shannon–Wiener diversity index (Shannon and Weaver, 1963; Pielou, 1966), as:

$$H' = -\Sigma fr_i \ln fr_i.$$

We calculated also the evenness index, as $e = H'/H'\text{max}$, where $H'\text{max}$ corresponded to $\ln(S)$.

Landscape heterogeneity as consequence to natural and human-induced processes (e.g., disturbances) may be indirectly assessed using uni-variate metrics of diversity (Magurran and McGill, 2011). Among them, the concept of β-diversity has been applied along environmental gradients and patterns of heterogeneity (Whittaker, 1972; Wiens, 1989; Koleff et al., 2003; Magurran, 2004; Dornelas et al., 2011). In our study, to assess the habitat heterogeneity of each habitat type, we calculated the Whittaker β_w index (Whittaker, 1960), as:

$$\beta_w = S/S_{\text{mean}},$$

where S was considered a measure of γ-diversity and S_{mean}, a measure of averaged α-diversity values among point counts. This index indirectly estimated a bird-related internal habitat heterogeneity assessing the species turnover among points (assuming that the higher the species turnover, the higher the habitat heterogeneity; Magurran, 2004).

In most of the ecological literature, the diversity/dominance curves are used as bi-variate approach to assess specific changes in the species assemblages (e.g., Whittaker plots; Whittaker, 1960; Battisti and Guidi, 2010 and Santoro et al., 2012). For instance, the profile, trend, and shape of the diagram lines may convey information on change in evenness and, consequently, in the assemblage structure (higher slope of tendency lines, higher stressed is an assemblage; Magurran, 2004). In these diagrams, all the species in a sample are ranked from the most to the least relatively abundant. Each species has a rank, which is plotted on the X-axis, and a frequency on the Y-axis. The frequency for the most abundant species is plotted first, then the next most common, and so on (see Magurran, 2004). In this work, we performed diversity/dominance diagrams comparing both the assemblages detected in control (MOSc) vs. treatment (MOSt) assemblages to detect possible differences in slope (z coefficient) of the tendency lines. We obtained the fit of the better curve calculating their coefficient of determination (R^2).

Table 1
Check-list of recorded bird species. Scientific name, level of conservation concern in national red list (Rondinini et al., 2013; see also Bulgarini et al., 1998; LIPU and WWF, 1999; VU: Vulnerable, NT: Near threatened, LT: Least concern; DD: Data deficient) and inclusion in the Annex 1 of 409/79/CEE Bird Directive ('1') are reported.

Species	Red list (national)	Bird Directive	Species	Red list (national)	Bird Directive
Phalacrocorax carbo	LC		*Saxicola torquatus*	VU	
Pernis apivorus	LC	1	*Turdus merula*	LC	
Milvus migrans	NT	1	*Turdus viscivorus*	LC	
Milvus milvus	VU	1	*Hippolais poliglotta*	LC	
Circaetus gallicus	VU	1	*Sylvia cantillans/subalpina*	LC	
Buteo buteo	LC		*Sylvia melanocephala*	LC	
Falco tinnunculus	LC		*Sylvia communis*	LC	
Falco vespertinus	VU	1	*Sylvia atricapilla*	LC	
Falco peregrinus	LC	1	*Phylloscopus sibilatrix*	LC	
Columba livia f. domestica	–		*Phylloscopus collybita*	LC	
Coturnix coturnix	DD		*Regulus ignicapilla*	LC	
Columba palumbus	LC		*Aegithalos caudatus*	LC	
Streptopelia turtur	LC		*Periparus ater*	LC	
Cuculus canorus	LC		*Cyanistes caeruleus*	LC	
Asio otus	LC		*Parus major*	LC	
Athene noctua	LC		*Sitta europaea*	LC	
Strix aluco	LC		*Certhia brachydactyla*	LC	
Caprimulgus europaeus	LC	1	*Oriolus oriolus*	LC	
Apus apus	LC		*Garrulus glandarius*	LC	
Merops apiaster	LC		*Pica pica*	LC	
Picus viridis	LC		*Corvus monedula*	LC	
Dendrocopos major	LC		*Corvus cornix*	LC	
Lullula arborea	LC	1	*Passer domesticus/italiae*	LC (*italiae*: VU)	
Alauda arvensis	VU		*Fringilla coelebs*	LC	
Hirundo rustica	NT		*Serinus serinus*	LC	
Anthus pratensis	NT		*Chloris chloris*	NT	
Troglodytes troglodytes	LC		*Carduelis carduelis*	LC	
Erithacus ribecula	LC		*Carduelis cannabina*	LC	
Luscinia megarhynchos	LC		*Emberiza cirlus*	LC	
Phoenicurus ochruros	LC		*Emberiza calandra*	LC	
Phoenicurus phoenicurus	LC		N	61	

Values of S_{mean} and ABB_{mean} for the two sub-habitat types were compared using the non-parametric Mann–Whitney U test. We performed a non-parametric two-sample Kolmogorov–Smirnov test to compare two set of data (pairs of frequency distributions in MOSc vs. MOSt) to determine whether they come from the same distribution (Dytham, 2011). Finally, we compared the relative frequencies of species and guilds in MOSc vs. MOSt using the χ^2 test (only for species having at least 5 records in total). We also correlated the frequency values among species using a non-parametric Spearman rank correlation test (2 tail).

All tests were two-tailed, and alpha was set at 5%. We used the softwares SPSS 13.0 for Windows (SPSS Inc. 2003) and Primer 4.2 (Glantz, 1996). Alfa was set at 0.05.

We checked for data reliability using the data-sheets in Battisti et al. (2014a,b). Taxonomic nomenclature followed AERC TAC (2003) and Fracasso et al. (2009).

3. Results

Totally, we detected 61 bird species (including also not breeding species: vagrants, long-distance migrants; Table 1). Among them, four species were included in the IUCN category as "Near Threatened" and 5 as "Vulnerable" (see Rondinini et al., 2013). Using the PCM, we obtained 286 records belonging to 27 breeding species (22 in MOSc and 20 in MOSt) (Table 2).

We did not observe significant differences between relative frequencies in MOSc and MOSt both at species (χ^2 test; Table 2) and guild level (open/edge vs. mosaic/forest species: $\chi^2 = 0.001$, $p = 0.977$).

Comparing the relative frequencies between MOSc and MOSt, we observed a direct and significant correlation ($r_s = 0.696$, $p = 0.000$; Spearman rank correlation test, 2 tail, $n = 27$).

At community level, differences in values both for ABB_{mean} and S_{mean} were not significant when comparing MOSc and MOSt sub-habitats (respectively, $Z = -1.100$, $p = 0.271$ and $Z = -0.639$, $p = 0.523$, Mann–Whitney U test; Fig. 1).

The values of H' index were apparently similar between the MOSc and MOSt habitat sub-types (Table 3). The β-diversity index did not show evident differences between MOSc and MOSt (Table 3).

We obtained higher values of coefficient of determination (always $R^2 > 0.80$) for the logarithmic curves (better fit; Fig. 2). The diversity/dominance diagrams (Whittaker plots) showed as the curves are largely overlapping when comparing the curves for treatment and control assemblages ($z = -0.07$ for both of them; not significant difference: $Z = 0.588$, $p = 0.879$; two-sample Kolmogorov–Smirnov test; Fig. 2).

Table 2
Detected common breeding bird species, total number of records (N) and their relative frequencies (Fr) in open habitats and oak mosaics and for the two sub-areas (control and treatment with windmills). Species are grouped in two main ecological guild (m/f: mosaic/forest species; o/e: open and edge habitats; see methods). The values of χ^2 and p are reported (test performed only for species with total $n > 5$).

Species	Control	Treatment	χ^2 and p	Total	
	Fr	*Fr*		*Fr*	*N*
Falco tinnunculus (o/e)		0.01		0.003	1
Columba palumbus (m/f)	0.022	0.038	0.175 (0.675)	0.028	8
Streptopelia turtur (m/f)		0.01		0.003	1
Cuculus canorus (m/f)	0.011	0.048	2.348 (0.125)	0.024	7
Apus apus (o/e)	0.006			0.003	1
Picus viridis (m/f)	0.006	0.019		0.01	3
Lullula arborea (o/e)	0.006			0.003	1
Erithacus rubecula (m/f)	0.061	0.086	0.310 (0.578)	0.07	20
Luscinia megarhynchos (m/f)	0.017	0.019		0.017	5
Turdus merula (m/f)	0.022	0.01		0.017	5
Turdus viscivorus (m/f)		0.01		0.003	1
Sylvia cantillans/Sylvia subalpina (o/e)	0.193	0.171	0.003 (0.956)	0.185	53
Sylvia communis (o/e)		0.01		0.003	1
Sylvia atricapilla (m/f)	0.243	0.219	0.006 (0.940)	0.234	67
Phylloscopus collybita (m/f)	0.006			0.003	1
Periparus ater (m/f)	0.006			0.003	1
Cyanistes caeruleus (m/f)	0.061	0.105	1.244 (0.265)	0.077	22
Parus major (m/f)	0.055	0.038	0.132 (0.716)	0.049	14
Certhia brachydactyla (m/f)	0.006			0.003	1
Oriolus oriolus (m/f)	0.033	0.01	0.722 (0.396)	0.024	7
Garrulus glandarius (m/f)	0.05	0.019	0.963 (0.326)	0.038	11
Corvus cornix (o/e)	0.017			0.01	3
Fringilla coelebs (m/f)	0.061	0.057	0.018 (0.893)	0.059	17
Carduelis carduelis (o/e)		0.019		0.007	2
Carduelis cannabina (o/e)	0.017			0.01	3
Emberiza cirlus (o/e)	0.077	0.048	0.557 (0.455)	0.066	19
Emberiza calandra (o/e)	0.028	0.057	0.869 (0.351)	0.038	11
	$N = 181$	$N = 105$			$N = 286$

Table 3
Structural parameters of breeding bird assemblages in oak mosaics (MOS tot) and in two sub-areas (MOSc: control; MOSt: treatment, with windmills). For abbreviations, see methods.

Parameters	MOSc	MOSt	MOS tot
S (γ)	22	20	27
S_{mean} (α_{mean})	4.15 (1.603)	3.90 (1.51)	4.056 (1.559)
N (ABB TOT)	181	105	286
ABB_{mean}	5.48 (2.002)	5 (1.817)	5.296 (1.929)
H'	2.502	2.526	2.561
e	0.81	0.84	0.77
β index	5.30	5.13	6.66

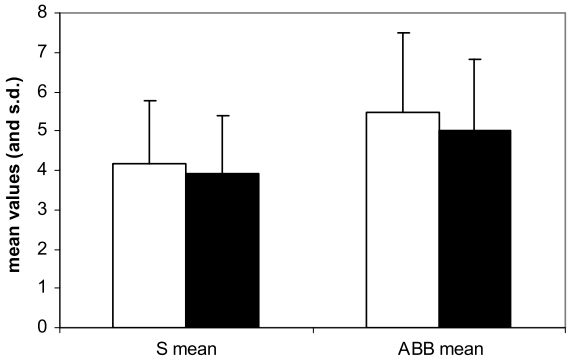

Fig. 1. Comparison between mean number of species (S_{mean}) and mean abundance index (ABB_{mean}) in sub-areas control (MOSc, white columns) and treatment (MOSt, black columns) of oak mosaic.

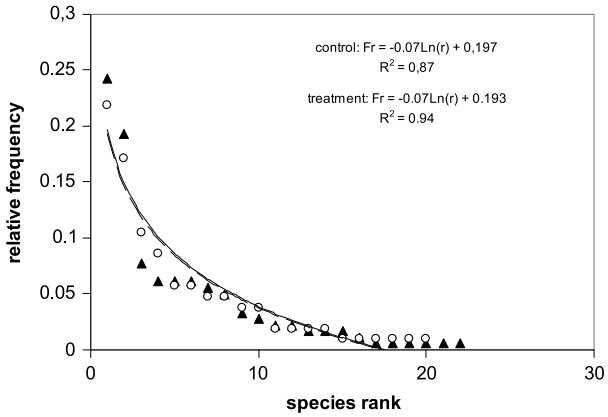

Fig. 2. Whittaker plot (species rank/relative frequency diagram) comparing the species assemblages in control (black triangles and continuous line) and treatment areas (white circles and dashed line). Equation of regression logarithmic line and coefficient of determination (R^2) for control (MOSc) and treatment (MOSt) sub-areas (oak mosaics) are reported.

4. Discussion

Bird communities are structured in response to vegetation and environmental constraints, mainly (vertical) vegetation structure and (horizontal) spatial heterogeneity (Whittaker, 1972; Wiens, 1989). In this sense, our study area hosted rich breeding bird assemblages typically characterizing the highly heterogeneous oak mosaic landscapes of the central Apennines (Santone and Di Carlo, 1994; Farina, 1997; Pellegrini et al., 2007). Totally, the number of species recorded (>60) was relatively high when compared to analogous patchy hilly landscapes (e.g. Frank and Battisti, 2005 and Lorenzetti and Battisti, 2006). This fact may be due to the high historical and recent human-induced spatial heterogeneity and structural complexity of these landscapes that increased the availability in niches and resources for many species (e.g. patchiness and presence of ecotones, etc.). Moreover this area (in particular ridges and valley) is also an important site of migration stop-over (Adriatic and South-eastern Mediterranean migratory route; Spina and Volponi, 2008).

Contrary to our predictions, we did not observe differences between oak mosaics hosting wind farm turbines (treatment sub-area) and non-impacted oak mosaics (control sub-area), also using a bi-variate diversity/dominance approach. Probably the low level of fine-grained perforation and dissection induced by windmills and their related infrastructures and roads, did not affect the response in common breeding birds at any investigated level (assemblage, guild and species level).

The location of wind farm turbines generally may induce habitat perforation (Bayne and Dale, 2011), and secondarily dissection by service roads, two specific spatially modelled-patterns of habitat fragmentation that affect bird population and communities (Helle, 1985; Robinson et al., 1992; Matthysen et al., 1995; Fahrig, 1997; Norton et al., 2000; Schmiegelow and Mönkkönen, 2002; Fischer and Lindenmayer, 2007). However, at present, the effect of habitat perforation have been studied only regarding to forest harvesting while research on the effects of perforation due to wind farm facilities are still lacking. For example, Leupin et al. (2004) showed as habitat perforation due to harvesting in high-elevation coniferous woods has not significantly changed the songbird community. However, the effects of perforation may differ, being influenced by total basal area of wood removed and by other factors (context, scale, grain of perforation: Robinson and Robinson, 1999).

Therefore, apparently the 10% in habitat perforation (30 ha on about 300 ha in size of oak mosaics) was not enough to disrupt the assemblage, guild and population structure, at least at the spatial and temporal scale analysed in this study. There are many evidences that the effects of habitat perforation are evident only when overcoming specific thresholds (Suarez-Rubio et al., 2013). More in general, in habitat fragmentation studies it has been highlighted as population and community collapses may develop starting at higher thresholds in habitat changes (see Fahrig, 1997), with responses that are highly context-, scale- and species-dependent (Fischer and Lindenmayer, 2007).

The β-diversity is a key concept for understanding the functioning of ecosystems, for the conservation of biodiversity and for ecosystem management (Legendre et al., 2005). The values of this metric highlights the level of species turnover among point counts and, therefore, the degree of internal habitat heterogeneity in a landscape (Magurran, 2004; Magurran and McGill, 2011). Using this diversity metric, inside the oak mosaics we did not observe differences between control and treatment sub-types (these last with windmills), both at assemblage and species level. The changes at patch level (i.e. internal at the mosaics) induced by the wind turbines, roads and other infrastructures, and to related disturbances (as a higher motorcar transit and people frequentation), apparently did not affect the common breeding bird communities at landscape level. In particular, the values of H' and β-diversity were very similar between MOSc and MOSt, suggesting that these assemblages perceived these two sub-types as having a similar level of patchiness, complexity and disturbance. The direct and significant correlation between relative frequencies confirmed these considerations. Probably, changes in heterogeneity due to wind farm were too limited in extension (i.e. they are below a threshold in area) and the related disturbances too low in magnitude to induce significant effects at the assemblage level (see Sousa, 1984; Pickett and White, 1985 and Brawn et al., 2001).

To our knowledge, this was the first research that tested the possible control-treatment changes in breeding bird populations, guilds and assemblages living in mosaics perforated by wind farm facilities. However, we highlighted some weaknesses of this study: (1) we tested possible control-treatment differences only in the same seasonal cycle, not investigating before-after changes at higher temporal scales; (2) we analysed only the structural patterns of populations, guilds and assemblages, not considering other complex medium- and long-term interactions, dynamics and consequences due to habitat perforation (e.g. increase of edge habitats and consequently nest predation: e.g. Huhta et al., 2004); (3) the point count method used has been focused only on a local set of common breeding birds and our conclusions on a lack of indirect impact of a windfarm were limited to this easily detectable species and a single case study; 4) our study was carried out in the season immediately following the placement of wind turbines in the site (winter 2013): in this sense our considerations on the absence of effects of wind turbines only apply to a short-term response of bird assemblage, guilds and species and we cannot exclude that structural changes at different levels may occur in medium-long term (lag effect; Masden et al., 2010; for fragmented and perforated landscapes: Kupfer and Franklin, 2009).

In this sense, since this landscape of the central and Southern Apennine shows a high ornithological interest (Important Bird Area IBA 115; Gariboldi et al., 2000; AA VV, 2003 and Teofili et al., 2009), we suggest a precautionary approach, so stimulating further studies at higher spatial and temporal scales and focused on more sensitive targets to direct collision (e.g. raptors of higher conservation concern, threatened species at national level; Percival, 2005; Drewitt and Langston, 2008; De Santis et al., 2009; Garvin et al., 2011; Saidur et al., 2011 and Rondinini et al., 2013). Finally, from an ecological perspective, we also encourage further studies focused on the effects on wind farm facilities on different ecologically-related guilds (e.g., insectivorous vs. granivorous, cavity nesters, etc.; Verner, 1984).

Acknowledgements

This study has been originally carried out to detect indirect impact of the San Giovanni Lipioni wind farm by Agrifolia and Studio Arcadia Society for Iberdrola Ingenieria y Costrucción—Eolica Lucana s.r.l. (act n. E 118 PEG EG 001 120704). Dr. Ph.D. Alessandro Zocchi reviewed the English language. The Editor and two anonymous reviewers largely improved a first draft of the manuscript with useful comments and suggestions.

References

AA. VV., 2003. Risoluzione in merito all'impatto degli impianti eolici sui Rapaci e sull'avifauna in genere. Atti I Convegno Italiano Rapaci Diurni e Notturni. Preganziol (TV). Avocetta 27, 150.
AERC TAC, 2003. AERC TAC Checklist of bird taxa occurring in Western Palearctic Region, with distributional notes on subspecies – 15th Draft on line: http://www.aerc.be.
Battisti, C., Dodaro, G., Franco, D., 2014a. The data reliability in ecological research: a proposal for a quick self-assessment tool. Nat. Hist. Sci. 1, 75–79.
Battisti, C., Fanelli, G., 2011. Does human-induced heterogeneity differently affect diversity in vascular plants and breeding birds? Evidences from three Mediterranean forest patches. Rend. Lincei 22, 25–30.
Battisti, C., Franco, D., Norscia, C., Santone, P., Soccini, C., Ferri, V., 2014b. Estimating the indirect impact of wind farms on breeding bird assemblages: a case study in the central Apennines. Isr. J. Ecol. Evol. 59, 125–129.
Battisti, C., Guidi, A., 2010. Comparing spatial rarity between long distance migrant and sedentary bird species using atlas data and diversity/dominance diagrams. Vie Milieu—Life Environ. 60, 321–326.
Bayne, E.M., Dale, B.C., 2011. Effects of energy development on songbirds. In: Naugle, D.E. (Ed.), Energy Development and Wildlife Conservation in Western North America. Island Press, Washington, DC, pp. 95–114.
Bellamy, P.E., Hinsley, S.A., Newton, I., 1996. Factors influencing bird species numbers in small woods in south-east England. J. Appl. Ecol. 33, 249–262.
Bibby, C.J., Burgess, N.D., Hill, D.A., Mustoe, S., 2000. Bird Census Techniques, second ed. Academic Press, London.
Birdlife International, 2004. Birds in Europe: Population Estimates, Trends and Conservation Status. In: Birdlife Conservation Series, vol. 12. Birdlife International, Cambridge, UK.
Blondel, J., Aronson, J., 1999. Biology and Wildlife of the Mediterranean Region. Oxford University Press, Oxford.
Brawn, J.D., Robinson, S.K., Thompson III, F.R., 2001. The role of disturbance in the ecology and conservation of birds. Annu. Rev. Ecol. Syst. 32, 251–276.
Brotons, L., Herrando, S., Martin, J.-L., 2005. Bird assemblages in forest fragments within Mediterranean mosaics created by wild fires. Landsc. Ecol. 19, 663–675.
Bulgarini, F., Calvario, E., Fraticelli, F., Petretti, F., Sarrocco, S., 1998. Libro rosso degli animali d'Italia. Vertebrate. WWF Italia, Rome.
Cieslak, M., 1985. Influence of forest size and other factors on breeding bird species number. Ekol. Pol. 33, 103–121.
De Santis, A., Allavena, S., Artese, C., 2009. What is the quality standard of the 'EIA' process for wind farms in the Abruzzo Region, central Italy? In: Proceedings of XV Italian Conference in Ornithology. Alula, Vol. 16, pp. 41–46.
Derleth, E.L., McAuley, D.G., Dwyer, T.J., 1989. Avian community response to small-scale habitat disturbance in Maine. Can. J. Zool. 67, 385–390.
Dornelas, M., Soykan, C.U., Ugland, K.I., 2011. Biodiversity and disturbance. In: Magurran, A., McGill, B.J. (Eds.), Biological Diversity. Frontiers in Measurements and Assessments. Oxford University Press, Oxford, New York, pp. 237–251.
Drewitt, A.L., Langston, R.H.W., 2008. Collision effects of wind-power generators and other obstacles on birds. Ann. New York Acad. Sci. 1134, 233–266.
Dytham, C., 2011. Choosing and Using Statistic. A Biologist's Guide. Wiley-Blackwell, UK.
Fahrig, L., 1997. Relative effects of habitat loss and fragmentation on population extinction. J. Wildl. Manage. 61, 603–610.
Fahrig, L., Baudry, J., Brotons, L., Burel, F.G., Crist, T.O., Fuller, R.J., Sirami, C., Siriwardena, G.M., Martin, J.L., 2011. Functional landscape heterogeneity and animal biodiversity in agricultural landscapes. Ecol. Lett. 14, 101–112.
Farina, A., 1997. Landscape structure and breeding bird distribution in a sub-Mediterranean agro-ecosystem. Landsc. Ecol. 12, 365–378.
Ferrer, M., de Lucas, M., Janss, G.F.E., Casado, E., Muñoz, A.R., Bechard, M.J., Calabuig, C.P., 2011. Weak relationship between risk assessment studies and recorded mortality in wind farms. J. Appl. Ecol. 49, 38–46.
Fischer, J., Lindenmayer, D.B., 2007. Landscape modification and habitat fragmentation: a synthesis. Glob. Ecol. Biogeogr. 16, 265–280.
Forman, R.T., 1995. Land Mosaics. The Ecology of Landscapes and Regions. Cambridge University Press, New York.
Forsman, J.T., Reunanen, P., Jokimäki, J., Mönkkönen, M., 2010. The effects of small-scale disturbance on forest birds: a meta-analysis. Can. J. For. Res. 40, 1833–1842.
Fracasso, G., Baccetti, N., Serra, L., 2009. La lista CISO-COI degli Uccelli italiani parte prima: liste A, B e C. Avocetta 33, 5–24.

Frank, B., Battisti, C., 2005. Area effect on bird communities, guilds and species in a highly fragmented forest landscape of Central Italy. Ital. J. Zool. 72, 297–304.
Gariboldi, A., Rizzo, V., Casale, F., 2000. Aree Importanti per l'Avifauna. LIPU, Parma.
Garvin, J.C., Jennelle, C.S., Drake, D., Grodsky, S.M., 2011. Response to raptors to a windfarm. J. Appl. Ecol. 48, 199–209.
Glantz, S.A., 1996. Statistics for Medical Disciplines. Version 4.2. McGraw and Hill, London.
Gove, B., Langston, R.H.W., McCluskie, A., Pullan, J.D., Scrase, I., 2013. Wind farms and birds: An updated analysis of the effects of wind farms on birds, and best practice guidance on integrated planning and impact assessment. Bern Convention, Strasbourg.
Helle, P., 1985. Effects of forest fragmentation on bird densities in northern boreal forests. Ornis Fenn. 62, 35–41.
Hinsley, S.A., Bellamy, P.E., Newton, I., Sparks, T.H., 1995. Habitat and landscape factors influencing the presence of individual breeding bird species in woodland fragments. J. Avian Biol. 26, 94–104.
Huhta, E., Aho, T., Jäntti, A., Suorsa, P., Kuitunen, M., Nikula, A., Hakkarainen, H., 2004. Forest fragmentation increases nest predation in the Eurasian Treecreeper. Conserv. Biol. 18, 148–155.
Huso, M.M.P., 2011. An estimator of wildlife fatality from observed carcasses. Environmetrics 22, 318–329.
Johnson, G.D., Erickson, W.P., 2010. Avian, bat and habitat cumulative impacts associated with wind energy development in the Columbia Plateau Ecoregion of Eastern Washington and Oregon. Klickitat County Planning Department, Western EcoSystems Technology, Inc.
Koleff, P., Gaston, K.J., Lennon, J.J., 2003. Measuring beta diversity for presence–absence data. J. Anim. Ecol. 72, 367–382.
Koskimies, P., Väisänen, R.A., 1991. Monitoring bird populations. Zoological Museum, Finland Museum of Natural History.
Kupfer, J.A., Franklin, S.B., 2009. Linking spatial pattern and ecological responses in human-modified landscapes: The effects of deforestation and forest fragmentation on biodiversity. Geogr. Compass 3, 1331–1355.
Langston, R.H.W., Pullan, J.D., 2003. Windfarms and birds: an analysis of the effects of windfarms on birds, and guidance on environmental assessment criteria and site selection issues. In: XXIII Meeting of Standing Committee, BirdLife International, RSPB, Strasbourg.
Legendre, P., Borcard, D., Peres-Neto, P.R., 2005. Analyzing Beta diversity: partitioning the spatial variation of community composition data. Ecol. Monogr. 75, 435–450.
Leupin, E.E., Dickinson, T.E., Martin, K., 2004. Resistance of forest songbirds to habitat perforation in a high-elevation conifer forest. Can. J. For. Res. 34, 1919–1928.
LIPU and WWF, 1999. Nuova Lista Rossa degli uccelli nidificanti in Italia. Riv. Ital. Ornitol.—Res. Ornithol. 69, 3–43.
Lorenzetti, E., Battisti, C., 2006. Area as component of habitat fragmentation: corroborating its role in breeding bird communities and guilds of oak wood fragments in Central Italy. Rev. Ecol. (Terre Vie) 61, 53–68.
Magurran, A., 2004. Measuring Biological Diversity. Blackwell Publishing, Malden.
Magurran, A., McGill, B.J., 2011. Biological Diversity. Frontiers in Measurements and Assessments. Oxford University Press, Oxford.
Martì, M.R.E., Barrios, J., 1995. Effects of Wind Turbine Power Plants on the Avifauna in the Campo de Gibraltar Region. Spanish Ornithological Society, Madrid.
Masden, E.A., Fox, A.D., Furness, R.W., Bullman, R., Haydon, D.T., 2010. Cumulative impact assessments and bird/wind farm interactions: developing a conceptual framework. Environ. Impact Assess. 30, 1–7.
Matthysen, E., Lens, L., Van Dongen, S., Verheyen, G.R., Wauters, L.A., Adriaensen, F., Dhondt, A.A., 1995. Diverse effects of forest fragmentation on a number of animal species. Belgian J. Zool. 125, 175–183.
Møller, A.P., 1987. Breeding birds in habitat patches: random distribution of species and individuals?. J. Biogeogr. 14, 225–236.
Moore, N.W., Hooper, M.D., 1975. On the number of bird species in British woods. Biol. Cons. 8, 239–250.
Morrison, M.L., 2002. Wildlife Restoration. Techniques for Habitat Analysis and Animal Monitoring. Island Press, Washington.
Noguera, J.C., Pérez, I., Minguez, E., 2010. Impact of terrestrial wind farms on diurnal raptors developing a spatial vulnerability index and potential vulnerability maps. Ardeola 57, 41–53.
Norton, M.R., Hannon, S.J., Schmiegelow, F.K.A., 2000. Fragments are not islands: patch vs landscape perspectives on songbird presence and abundance in a harvested boreal forest. Ecography 23, 209–223.
Opdam, P., Rijsdijk, G., Hustings, F., 1985. Bird communities in small woods in an agricultural landscape: effects of area and isolation. Biol. Cons. 34, 333–352.
Pellegrini, M., Antonucci, A., Artese, C., Carafa, M., Cirillo, M., De Sanctis, A., Dundee, V., Lalli, G., Strinella, E., 2007. Check-list degli uccelli d'Abruzzo. Riv. Ital. Ornitol.—Res. Ornithol. 77, 27–38.
Percival, S., 2005. Birds and windfarms: what are the real issues? Br. Birds 98, 194–204.
Petrella, S., Bulgarini, F., Cerfolli, F., Polito, M., Teofili, C., 2005. Libro rosso degli habitat della Rete Natura 2000. WWF Italia, Rome.
Pickett, S.T.A., White, P.S. (Eds.), 1985. The Ecology of Natural Disturbance and Patch Dynamics. Academic Press, New York.
Pielou, E.C., 1966. The measurement of diversity in different types of biological collections. J. Theoret. Biol. 13, 131–144.
Pirone, G., 1995. Trees, Bushes and Lianas of Abruzzo. Cogecstre Edizioni, Penne, Pescara.
Richardson, W.J., 2000. Bird migration and wind turbines: migration timing, flight behaviour, and collision risk. In: Proceedings of National Avian—Wind Power Planning Meeting II, pp. 132–140.
Robinson, G.R., Holt, R.D., Gaines, M.S., Hamburg, S.P., Johnson, M.L., Fitch, H.S., Martinko, E.A., 1992. Diverse and contrasting effects of habitat fragmentation. Science 257, 524–526.
Robinson, W.D., Robinson, S.K., 1999. Effects of selective logging on forest bird populations in a fragmented landscape. Conserv. Biol. 13, 58–66.
Rondinini, C., Battistoni, A., Peronace, V., Teofili, C., 2013. Lista Rossa IUCN dei vertebrati italiani. In: Comitato italiano IUCN e Ministero dell'Ambiente e della Tutela del territorio e del Mare, Rome.
Saidur, R., Rahim, N.A., Solangi, K.H., 2011. Environmental impact of wind energy. Renewable Sustainable Energy Rev. 15, 2423–2430.
Santone, P., Di Carlo, E.A., 1994. Check-list degli Uccelli d'Abruzzo e Molise. Gli Uccelli d'Italia 19, 25–38.
Santoro, R., Jucker, T., Prisco, I., Carboni, M., Battisti, C., Acosta, T.R., 2012. Effect of trampling limitation on coastal dune plant communities. Environ. Manag. 49, 534–542.
Schmiegelow, F.K.A., Mönkkönen, M., 2002. Habitat loss and fragmentation in dynamic landscapes: avian perspectives from the boreal forest. Ecol. Appl. 12, 375–389.
Shannon, C.E., Weaver, W., 1963. Mathematical Theory of Communication. University of Illinois Press, Urbana, Illinois.
Sorace, A., Gustin, M., Calvario, E., Ianniello, L., Sarrocco, S., Carere, C., 2000. Assessing bird communities by point counts: repeated sessions and their duration. Acta Ornithol. 35, 197–202.
Sousa, W.P., 1984. The role of disturbance in natural communities. Annu. Rev. Ecol. Syst. 15, 353–391.
Spina, F., Volponi, S., 2008. Atlante della Migrazione degli Uccelli in Italia. Istituto Superiore per la Protezione e la Ricerca Ambientale, Rome.
Suarez-Rubio, M., Wilson, S., Leimgruber, P., Lookingbill, T., 2013. Threshold responses of forest birds to landscape changes around exurban development. PLoS One 8 (6), e67593.
Sutherland, W.J., 2006. Ecological Census Techniques. Blackwell Science, Massachussets.
Teofili, C., Petrella, S., Varriale, M., 2009. Valutazione di idoneità ecologica per la realizzazione di impianti eolici industriali in Italia. In: Proceedings of XV Italian Conference in Ornithology, Alula, Vol. 16, pp. 53–58.
Ukmar, E., Battisti, C., Luiselli, L., Bologna, M.A., 2007. The effects of fire on communities, guilds and species of breeding birds in burnt and control pinewoods in central Italy. Biodivers. Conserv. 16, 3287–3300.
Verner, J., 1984. The guild concept applied to management of bird populations. Environ. Manag. 8, 1–14.
Whittaker, R.H., 1960. Vegetation of the Siskiyou Mountains, Oregon and California. Ecol. Monogr. 30, 279–338.
Whittaker, R.H., 1972. Evolution and measurement of species diversity. Taxon 21, 213–251.
Wiens, J.A., 1989. The Ecology of Bird Communities. Vol. 2. Processes and Variations. In: Cambridge Studies in Ecology, Cambridge University Press, Cambridge.

Williams, T.C., Williams, J.M., Williams, P.G., Stokstad, P., 2001. Bird migration through a mountain pass studied with high resolution radar, ceilometers, and census. Auk 118, 389–403.
Winkelman, J.E., 1992. The impact of the Sep wind park near Oosterbierum (Fr.), the Netherlands, on birds, 1: collision victims. DLO Instituut voor Bos-en Natuuronderzoek. RIN-rapport 92/3 4 volumes.
Winkelman, J.E., 1995. Bird/wind turbine investigations in Europe. In: Proceedings National Avian-Wind Power Planning Meeting. Denver, Colorado, pp. 110–140.
Zamora, J., Verdú, J.R., Galante, E., 2007. Species richness in Mediterranean agro-ecosystems: Spatial and temporal analysis for biodiversity conservation. Biol. Cons. 134, 113–121.
Zieliński, P., Bela, G., Marchlewski, A., 2009. Report on monitoring of the wind farm impact on birds in the vicinity of Gnieźdźevo (gmina Puck, pomorskie voivodeship). Available online: www.proj6.turbo.pl/upload/file/343.pdf.